chapter thirty-one

Restoration of ponderosa pine forests in the interior western U.S. after logging, grazing, and fire suppression

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31.1 Introduction

Ponderosa pine forests are a major forest type found in every state in the western U.S. They are readily accessible and aesthetically appealing, and they have provided valuable economic benefits from logging and grazing. Human activities have contributed to

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extensive alteration of these forests. While early Native American effects may have been locally significant (Allen 2002; Allen et al. 2002; Baker and Kipfmueller 2001; Barrett and Arno 1982; Kaye and Swetnam 1999), most impacts in much of the western U.S. have probably occurred during the last 130–150 years, since settlement by Euro-Americans.

In many western U.S. regions, early photographs, journals written by explorers and surveyors, and scientific reports supplement physical evidence of historical forest structure. Physical evidence includes stand remnants of historical landscapes, fire scars providing insight into a major stand regulating influence, and residual stumps from early logging. Together, these sources of information provide considerable insight into the structural characteristics of natural ponderosa pine forests and their ecology prior to significant human influence (Moore et al. 1999; Swetnam et al. 1999).

Three varieties of ponderosa pine occur. These are Pacific ponderosa pine (*Pinus ponderosa* var. *ponderosa*) along the West Coast (particularly Oregon and Washington), Arizona ponderosa pine (*Pinus ponderosa* var. *arizonica*) found in southern Arizona and New Mexico, and the Rocky Mountain variety (*Pinus ponderosa* var. *scopulorum*) in the interior western states (Oliver and Ryker 1990). Jeffrey pine (*Pinus jeffreyi*), found in California and Mexico, closely resembles ponderosa pine but is a distinct species. Studies of packrat middens indicate that Rocky Mountain ponderosa pine first occurred in northern New Mexico and Arizona about 10,000 to 12,000 years ago, central Colorado only about 5,500 to 7,000 years ago, and northern Wyoming and western Montana only about 2,000 years ago, suggesting a postglacial northward migration (J. Betancourt, U.S. Geological Survey, pers. comm.).

Ponderosa pine generally occurs at the lower elevational life zone of western forests, above drier nonforest vegetation types such as grasslands, shrublands, and pinyon–juniper woodlands, but below subalpine forests. Because the geographic area naturally occupied by ponderosa pine is very large, significant regional differences exist in climate, soils, and topography where ponderosa pine occurs (Oliver and Ryker 1990). In many areas of Arizona, Washington, Oregon, and California, and in other areas with sufficient precipitation, site productivity is higher than in most of the interior Rocky Mountains where precipitation is lower.

Historically, wildfire was a keystone regulating process in the ponderosa pine zone (Swetnam et al. 1999). Ponderosa pine often dominated this zone because it is considerably better adapted to wildfire than other tree species climatically suited to the same zone. Traits associated with fire resistance include open crowns, self-pruning branches, and thick relatively inflammable bark (Keeley and Zedler 1998; Graham 2003). Thick scales protect buds, and tight needle bunches enclose and protect meristems but open into a loose arrangement that does not favor combustion or propagation of flames. The foliage generally has high moisture content, and the roots are deep. These fire adaptations favored ponderosa pine over fire-sensitive species lacking these features, and wildfire often allowed pure or nearly pure ponderosa pine stands to exist. Fires sometimes damaged but did not kill trees, and the study of fire scars provides valuable insight into the role of past fires in the ecology of these forests (Swetnam and Baisan 1996; Agee 1998).

Human effects on ponderosa pine ecosystems include loss of old-growth forests through logging, introduction of exotic and often noxious weed species, damaging effects of overgrazing, and alteration of natural disturbance patterns such as fire and insect activity. Many forests that had been pure or nearly pure ponderosa pine now include competing tree species such as Douglas fir (*Pseudotsuga menziesii*) or piñon pine (*Pinus edulis*) and Rocky Mountain juniper (*Juniperus scopulorum*) (Biondi 1996). Nearly all ponderosa pine forests are far denser and more vulnerable to crown fire than they were historically. Thus, there is a clear need for forest restoration, both on publicly and privately owned lands.

In this chapter, we begin by summarizing the natural range of variability in stand structure, landscape characteristics, and disturbance regimes of ponderosa pine forests prior to Euro-American settlement. Next, we examine the changes in these forests that have occurred in the last 150 years. Finally, we explore restoration of sustainable ponderosa pine ecosystems, and cite four specific case studies from across the Rocky Mountain region. We focus on Rocky Mountain ponderosa pine forests, found from western Montana to Arizona and east through the Rocky Mountains and into the Great Plains. The interior variety occurs in the elevational zone between grasslands, desert scrub, or piñon–juniper forests at lower elevations and mixed conifer or subalpine forests at higher elevations. However, much of the content of this chapter is applicable for the Pacific and Arizona varieties and to Jeffrey pine.

31.2 Historical fire regimes, stand structures, and landscape characteristics

31.2.1 Variation in fire regimes and stand structures

While fire historically maintained ponderosa pine forests throughout its range, regional differences in fire regimes had significant effects on the structure of historical forest stands and landscapes. Three categories of factors affect how a fire behaves: fuels or vegetative structure, topography and other physical site characteristics, and weather. Relatively subtle differences in ponderosa pine stand development exist across the geographic range of the species, but in combination with physical site characteristics and weather, significant contrasts exist in fire behavior patterns and assemblages of stands into a landscape structure. In this chapter, we recognize two primary types of historical fire regimes in ponderosa pine forests: surface-fire regimes, in which fires burned at generally low to moderate intensity throughout a stand and caused little canopy mortality; and mixed-severity fire regimes, in which fires burned at variable intensity, producing patches where canopy mortality was nearly complete as well as patches of low canopy mortality.

Regional differences in historical fire behavior probably stemmed in large part from differences in understory productivity and fuel accumulation, which are influenced by precipitation and its seasonality, and crown base height, which may have been influenced by surface fire frequency and flame heights and by site productivity. In the southwestern U.S., frequent low-intensity surface fires were common in historical ponderosa pine forests, with mean fire intervals often less than 10 years (Swetnam and Baisan 1996; Brown et al. 2001; Allen et al. 2002). Stand-replacing fires were rare (Woolsey 1911; Cooper 1960), and tree mortality was more common among seedlings and small saplings than in larger trees. In the southwestern U.S., summer monsoon moisture flow provides enough rainfall for high understory productivity. When dry, the grass and forb biomass provided a continuous fuelbed for the spread of surface fire, particularly following wet El Niño events (Swetnam and Betancourt 1998). Trees grew taller in these sites, and self-pruning of lower branches (perhaps enhanced by surface fires) resulted in high crown base heights that limited the likelihood of crown fires (Figure 31.1). Furthermore, the frequent surface fires minimized the establishment of seedlings, which would have provided subcanopy fuel ladders to move fire from the surface into the canopy.

Fires were less frequent farther north in the southern and central Rocky Mountains, where intervals between landscape-scale fires often exceeded 40 or even 50 years (Goldblum and Veblen 1992; Brown and Seig 1996; Brown et al. 1999, 2000; Brown and Shepperd 2001; Kaufmann et al. 2001; Donnegan et al. 2001). Frequently, fires in these more inland areas were mixed in severity, having a nonlethal surface fire component intermixed with a lethal component that killed the overstory, often in a patchy pattern. Surface fuels may have been inadequate to spread surface fires during most years. In the absence of frequent surface fires, shrub communities often developed in the open forests, and fuels

Figure 31.1 Historical low-density ponderosa pine forest in the southwestern U.S. (U.S. Forest Service file photo).

accumulated beneath the crowns of trees. Furthermore, the crown base height was lower, and in the absence of surface fires small trees probably were more common and created fuel ladders. This resulted in portions of forests becoming vulnerable to crown fires, and in occasional periods of higher precipitation, grasses and forbs may have provided enough surface fuels for fire spread. The result was mixed-severity fires, with portions spreading only on the surface, and in other places killing some trees either by torching or scorching of individual trees or by localized crown fires (Figure 31.2). But in some places, fire completely killed all trees and created new openings having no trees. The type of fire behavior at any site undoubtedly depended on local understory and overstory fuel conditions, topography, and weather. Evidence suggests that the proportions of each fire effect in mixed-severity fires varied widely (Brown et al 1999; Kaufmann et al. 2000a; Veblen et al. 2000; Huckaby et al. 2001). Furthermore, these fires historically burned through much of the year, which also would contribute to variable fire effects.

The surface-fire regime was more common in the southwest, and the mixed-severity fire regime occurred more commonly in the eastern and northern portions of the Rocky Mountain ponderosa pine range. However, local variation in fire regimes undoubtedly occurred, and the transition geographically from surface to mixed-severity fire regimes was probably gradual across latitude and elevation. There is an analytical difficulty in sorting out fire regimes. Required data include not only evidence of fire frequency and mean fire return interval under historical conditions but also evidence of tree ages and spatial patterns of fires and tree survival. The ages of trees help determine specific locations within fire perimeters where fire was stand-replacing and where it was not. Without this critical information, it is often difficult to determine the actual historical fire regime governing landscapes over time. No evidence of a true crown-fire system (Romme and Despain 1989; Turner and Romme 1994) has been found in ponderosa pine-dominated landscapes, except for one study that suggested extensive 19th-century crown fires in part of the Black Hills, SD (Shinneman and Baker 1997).

31.2.2 Historical landscape structure

The surface and mixed-severity fire behavior patterns, in combination with regional differences in site productivity, resulted in very different landscape structures. More uniform forest landscapes developed where surface fires occurred frequently, particularly in areas

Figure 31.2 Historical forest structure in December 1896 after a mixed severity fire in 1880 (from Denver Water photo archives).

with a relatively flat topography (Figure 31.1). These forests were generally low in density and dominated by large overstory trees. Forest stands were locally patchy with small clumps of trees and many small openings (generally considerably less than one ha). At a larger scale, landscapes may have included larger openings that were relatively stable over long periods, perhaps influenced by local variations in soil characteristics (Weaver 1951; Cooper 1960; White 1985; Covington and Moore 1994; Fulé et al. 1997; Fulé et al. 2002a).

In contrast, more complex and dynamic forest landscapes developed in regions where the mixed-severity fire behavior pattern predominated (Figure 31.2). Denser patches of trees may have existed but were not extensive, and open woodlands and openings with no trees were common. Landscapes were patchy and heterogeneous, and periodic fires and sporadic tree recruitment into openings or open woodlands contributed to constant changes in landscape structure over time.

31.3 Current condition of ponderosa pine forests

Ponderosa pine forests occur in relatively accessible terrain, and winter snow packs usually do not limit year-round access. Accordingly, pine forests throughout the west were widely exploited soon after Euro-American settlement, and more than a century of logging and grazing, coupled with nearly a century of fire suppression, resulted in a very different forest structure than occurred historically in most areas (Kaufmann et al. 1992; Covington and Moore 1994). Fire exclusion began in most areas with the onset of extensive, unregulated livestock grazing in the late 1800s, apparently because the animals consumed the fine herbaceous fuels that formerly carried surface fires (Allen et al. 2002). Aggressive fire suppression policies instituted in the early 1900s resulted in continued exclusion of surface fires (Pyne 1982). Logging removed many of the larger and older trees and created an environment for germination and establishment of many new trees (Veblen and Lorenz 1991; Covington and Moore 1994; Kaufmann et al. 2000a; Friederici 2003). Livestock grazing, particularly cattle, reduced understory competition for seedlings and reduced fuels for the spread of surface fire, which further favored germination and survival of new seedlings (Belsky and Blumenthal 1997; Baisan and Swetnam 1997). Examples of forest condition prior to logging and change over time caused by logging and fire suppression are shown in Figure 31.3 and Figure 31.4. Fire suppression removed the major natural disturbance that historically kept tree density low, and large tree regeneration events occurred during climatically favorable periods (Savage et al. 1996). Together, these practices resulted in dense, relatively uniform stands and landscapes dominated by young, small trees.

During the 20th century, millions of ha of ponderosa pine forest developed much higher stand densities and canopy cover, and they became vulnerable to catastrophically large crown fires and damage to watersheds, and massive bark beetle epidemics. For

Figure 31.3 Example of urban encroachment into ponderosa pine forest, creating a wildland/urban interface zone in which property and lives are at risk of fire (photograph by David Steinke, USDA Forest Service).

Figure 31.4 Current condition of ponderosa pine forest on a north aspect near Cheesman Lake in the Colorado Front Range (photo by M.R. Kaufmann).

example, in the Colorado Front Range on the east face of the southern Rocky Mountains, a series of large wildfires occurred since 1996, most exceeding 4,000 ha, each having a large crown-fire component, and each well outside the range of fire severity that occurred historically (Brown et al. 1999; Graham 2003; M. Kaufmann, unpublished data). The 55,000-ha Hayman fire in June 2002 burned the Cheesman Lake landscape, a major research site for studies on historical ponderosa pine landscape characteristics (Brown et al. 1999; Kaufmann et al. 2000b, 2001, 2003; Huckaby et al. 2001; Fornwalt et al. 2002, 2003). Many of these forests also had large amounts of fuel created by a mountain pine bark beetle epidemic in the 1970s, exacerbated by high stand densities and tree crowding.

Where surface-fire regimes occurred historically, the recent crown fires are highly atypical (Woolsey 1911; Weaver 1951). Where mixed-severity fires occurred historically, the current crown-fire patch size appears to be considerably larger than occurred historically because of a greater accumulation of fuels in the past century or more (Kaufmann et al. 2000b, 2001). In both the surface and mixed-severity fire systems, large crown fires are not consistent with ecological sustainability. They result in large areas of charred forest that are vulnerable to excessive erosion and invasion by exotic plant species (Griffis et al. 2001), and they are unsuited for native species that favor historical forest structures. In addition, human population growth and urban encroachment into forests have placed people's lives and property at risk (Figure 31.3). More than half of the forest in many western counties is classified as a wildland/urban interface.

31.4 Restoration goals for ecologically sustainable landscapes

Large crown fires in ponderosa pine forests reset the forest landscape to an unforested condition. Reestablishment of older forests takes at least several centuries, as the old-growth condition does not occur until ponderosa pine trees are about 200 years old (Kaufmann 1996), and the oldest trees in unlogged landscapes are often more than 400 or even 600 years of age (Huckaby et al. 2001). Re-formation of soils and recovery of genetic diversity takes even longer; thus, the negative effects of large crown fires persist for a long time. In contrast, carefully designed forest treatments offer opportunities to improve forest structure, composition, and processes while reducing the likelihood of an ecological reset by large crown fire.

Differences in fire effects of the two natural fire behavior systems described above (surface and mixed-severity fire regimes) provide a basis for different approaches in forest restoration across the range of interior ponderosa pine forests. Where surface fire predominated historically, restored forests would be relatively uniform at a large scale, with fine-scale patchiness (on the order of tenths of a ha) provided by irregular tree distribution and small openings. Where mixed-severity fires occurred, however, the scale of patchiness would be larger (one to tens or hundreds of ha), reflecting the thinning or stand-replacing components of fire and irregular spatial and temporal patterns of tree establishment (Huckaby et al. 2001; Kaufmann et al. 2000a; Kaufmann et al. 2003). Because of differences in historical fire effects on landscape structure, no single set of management recommendations for restoration is appropriate across the entire range of ponderosa pine. Other differences, such as soil characteristics, patterns of nonfire disturbance, climate, or topography, also vary, underscoring the importance of site-specific restoration approaches. Finally, management goals and social issues commonly preclude "strict restoration" practices that are based exclusively on ecological factors (Higgs 1997).

It is noteworthy that restoring forests to regionally specific conditions similar to those occurring historically would address both the sustainability and wildfire hazard issues. Where properties or other human values are at risk, thinning forests and creating fuel breaks having a very low overstory density could reduce wildfire hazards. Such thinning would also lead to improved ecological sustainability where surface fire predominated historically. The restoration of a more complex landscape structure where mixed-severity fires occurred would also require thinning, but restoring dispersed patches of open woodlands and openings would be necessary to provide an ecologically sustainable landscape. A restored complex landscape would also provide considerable reduction in wildfire hazard. Therefore, forest restoration can mitigate wildfire hazards while also achieving ecological goals appropriate for each region.

Spatial scale is important. Forest restoration can be accomplished at nearly any scale, from stand to landscape. Even small-scaled restoration activities near a house may provide ecological benefits. However, landscape restoration requires that forest conditions be improved at a spatial scale of landscape processes such as fire or maintenance of viable species populations. Landscape restoration necessarily is a huge undertaking with many potential social, economic, and ecological issues and impediments.

Ideally, ponderosa pine restoration would yield forests having appropriate tree age and size distributions, properly configured spatially across the landscape to support a trajectory of future self-regulation of the landscape. Historically, ponderosa pine forests had a high percentage of old-growth trees, even in complex landscapes where mixed-severity fires often killed trees and created openings (Huckaby et al. 2001). Many current forests still have at least some old-growth trees, easily recognized by characteristic features of their tree crowns and branches (Kaufmann 1996; Huckaby et al. 2003). This aids in restoring the appropriate tree age distribution in forest landscapes (Mast et al. 1999). Large, young trees approach the size of old trees found in historical forests (Kaufmann et al 2000b; Allen et al. 2002). Where old trees are rare, these trees are candidates for retention. However, retention of all large trees ultimately leads to overstocked stands and increased wildfire hazard in some stands, and where openings should be re-created, such trees clearly must be removed. The number of smaller trees usually far exceeds historical densities, however, and small trees contribute significantly to current wildfire hazards.

31.5 Restoration treatment options

Forest restoration at any scale requires a careful consideration of the result desired and of the social acceptability of restoration alternatives. Land managers have three basic options: continue current fire suppression policies with no other treatment to forests; attempt to restore more sustainable forest structure and dynamics by means of natural or prescribed fire only; and actively restore more natural stand and landscape structures by means of mechanical thinning, prescribed fire, and other techniques. The first two options probably are not feasible. While leaving forests alone or using fires without mechanical thinning is appealing for certain ecological and social reasons (Cole 2001), fire suppression is simply not reliable enough to protect forests from fire. Thus, relying on fire suppression to hold ponderosa pine landscapes in their current condition without attempting restoration is difficult and hazardous.

In certain remote wildlands, a burn-only approach to restoration may be useful (Miller and Urban 2000). In a vast majority of ponderosa pine landscapes, however, high tree densities and biomass have already resulted in catastrophic crown fires well outside the historical patterns. Thus, the best alternative both ecologically and socially is the third option: manipulate ponderosa pine forest structure to align it with conditions that occurred naturally, thereby placing these forests on a trajectory of future self-regulation. Many agencies are carrying out fuel hazard and wildland fire use projects that involve the combination of thinning and prescribed fire, although the scale of the work is inadequate.

Two major types of treatment, mechanical thinning accompanied by removal of biomass, and the use of fire to reduce fuels, have proven useful for reducing forest density and biomass. In many ponderosa pine forests, there are 10 to 20 or more times as many trees as there were historically (Kaufmann et al. 2001). Compare a historical forest in the Colorado Front Range (Figure 31.2) with a current forest only 2 km away (Figure 31.4), for example. Felling trees but leaving slash in the forest does not appear to be a viable option. Subsequent fires are too intense and generally kill the retained trees. Converting trees into mulch with grinding machines is another option considered below.

Thinning from below in forests that historically had a surface fire regime can reduce the density of smaller trees while retaining both old trees and an appropriate diameter class distribution. Where mixed-severity fires occurred, some forest patches require thinning, but in other areas treatment should lead to either openings (complete clearcut) or open woodlands, both of which remove significant numbers of overstory trees regardless of size or even age. In a Colorado Front Range ponderosa pine landscape, 42% of the land area was estimated to have a historical canopy cover of 10% or less, 40% had between 11 and 30%, and only 7% had 35% canopy cover or higher (Kaufmann et al. 2001; Fornwalt et al. 2002; M. Kaufmann, unpublished data). In contrast, only 21% of current forests have a canopy cover of 10% or less, and 45% have a canopy cover of 35% or higher. At canopy covers of 35% or higher, crown fire occurs readily when other conditions are favorable for fire.

Prescribed fire is used to reduce wildfire hazard by reducing forest biomass, especially fine fuels and small trees (Figure 31.5). By choosing conditions under which to use prescribed fire, forest managers can effectively reduce wildfire hazards in many cases, and in rare instances may be able to recreate small openings or low-density patches by mortality of the overstory. However, prescribed burning by itself rarely can accomplish the changes in forest structure needed to make forests ecologically sustainable and safe from catastrophic fire. In many forests, a dense understory of smaller trees makes prescribed burning too likely to progress into a catastrophic crown fire. Furthermore, prescribed fire consumes deep litter accumulated beneath older trees, often producing enough heat to damage roots and kill trees that should be retained (Swezy and Agee 1991; Sackett et al. 1996). In general, it is difficult to remove enough biomass with prescribed fire, or to achieve the appropriate patch and forest structure needed to restore the landscape.

The use of both mechanical thinning and prescribed fire is very promising, however. Thinning can be used to reduce overstory density and canopy closure to levels found

Figure 31.5 Use of prescribed fire in Arizona to reduce understory fuels.

historically, thereby reducing the likelihood of active crown fire (fire spread from crown to crown). One additional option is the use of machines that chop or grind entire trees into chunks and scatter them in the forest. This process converts living biomass to material scattered on the ground, leaving it in a form that can be consumed with prescribed fire but not resulting in flame heights that lead to crown fires. However, such tree mulching is costly and has no potential for economic return, and at least temporary shifts in site nitrogen balance are likely.

Prescribed fire is useful for consuming slash and small trees left after mechanical treatment operations. Furthermore, prescribed burning is a primary tool for treatment in subsequent years, and in inaccessible areas. Frequently, soil disturbance during mechanical treatments and more open stand conditions creates favorable conditions for reproduction. In a few decades, treatment benefits are quickly lost. Prescribed fire may be an effective tool for limiting ingrowth after mechanical treatment. In areas having a historical surface-fire regime, relatively frequent prescribed fire could be used to mimic the historical fire effects. In areas having a mixed-severity fire regime, prescribed fire might be timed to kill new seedlings a few years after mechanical treatment, but subsequent prescribed fire may not be required for several decades.

31.6 The concept of a landscape strategy

Millions of ha of ponderosa pine forests are in poor ecological condition and pose high risk to human lives and property; yet on an annual basis, restoration treatments involving mechanical thinning are applied to only thousands of ha. Given the staggering economic and human resources needed for large-scale restoration, a strategy for implementing restoration in stages is needed to optimize the use of limited resources. This strategy should provide the largest ecological and wildfire mitigation benefits over the greatest amount of land in a reasonable time and at a low cost per ha, while minimizing social disruption. Considerable information is available to develop at least parts of such a strategy. Like adaptive management, the development of a restoration strategy should be considered an iterative process that begins with locally relevant scientific information for developing the desired ecological outcome.

The effect of fire behavior in the landscape is one strategic factor. For wildfire management, there is growing scientific evidence that under some circumstances, treating only a portion of a landscape in an appropriate treatment pattern may significantly alter crown fire spread for areas much larger than those treated (Finney 2001). Thus, carefully positioning fuel break treatments in areas having highly erosive soils or where critical habitats need protection may leverage treatment benefits to larger areas. More information of the ecological consequences of alternative treatment patterns across the landscape is needed.

Similarly, mechanical thinning treatments may be made less costly by positioning them in areas already lower in tree density or that are more accessible, and optimizing the use of less costly prescribed fire in concert with mechanical treatments may expand treatment benefits to larger areas at reduced cost. Furthermore, some trees may have economic value as forest products.

31.7 Case examples

Several ongoing projects in Arizona, Colorado, and Montana illustrate that forest restoration of ponderosa pine forests at a landscape scale can be accomplished while accommodating differences in regional fire behavior patterns and in the importance of issues at a local scale. Several features are common among all the case studies and are summarized here. First, all sites have suffered ecologically from the effects of logging, grazing, and fire suppression that led to dense and fairly uniform forests dominated by young, small-diameter trees (Figure 31.3 and Figure 31.4). Second, all are highly vulnerable to large crown fires and are ecologically unsustainable in their present condition. Third, each site had previously provided adequate material to support forest products industries, but industry presence is now limited or nonexistent. Fourth, all sites provide a range of other amenities, including hunting, firewood gathering, and general recreation.

While similar in many ways, these sites vary in other features, such as the degree of urban encroachment, role as municipal watersheds, concern about endangered and invasive species, and especially the approaches taken by partnerships to achieve improvement in ecological condition and reduction in wildfire hazard. They also differ in the potential to reestablish a viable industry that can assist in restoration activities through forest product markets. The case studies below are presented in order of increasing latitude from Arizona to Montana, and generally reflect a transition from surface fire regimes to mixed-severity fire regimes.

31.7.1 Northern Arizona, U.S.A. — including the Greater Flagstaff Forests Partnership

The high country of Arizona and New Mexico supports nearly 3 million ha of ponderosa pine and lower mixed conifer forests (Fiedler et al. 2002; O'Brien 2002), distributed among several national forests, national parks, and Native American lands. In New Mexico, over 90% of these forests are considered at moderate or high risk of crown fire due to dense stand structure and accumulated fuels (Fiedler et al. 2002). A long legacy of research has shown that the exclusion of frequent surface fires changed southwestern pine forests from open to dense conditions, dominated by fuel ladders of small, crowded trees (Weaver 1951; Cooper 1960; Covington and Moore 1994; Swetnam et al. 1999). The consequences of increased fuels may have been masked by unusually wet conditions from the mid-1970s to mid-1990s (Salzer in press), but a drought period beginning in 1996 has led to large and severe wildfires such as the Rodeo-Chediski fire of 2002, which covered approximately 190,000 ha. Recent landscape-scale bark beetle outbreaks, previously not seen in the southwest (Sánchez-Martínez and Wagner 2002), may further exacerbate fire behavior.

A pioneering attempt at restoring forest structure and function was initiated in Arizona in 1992 by Covington et al. (1997). Treated areas showed significant improvements in old-growth tree photosynthesis, reduced moisture stress, increased insect resistance (Feeney et al. 1998; Stone et al. 1999), higher understory plant productivity (Covington et al. 1997), and altered nutrient transformations and hydrological responses (Kaye and Hart 1998; Kaye et al. 1999). This set of findings provides a strong scientific basis for viewing treatments as restoring ecosystem-level patterns of structure and function, not simply as ameliorating fuel hazards.

The largest restoration project to date, on Bureau of Land Management lands in northwestern Arizona, covers 1,200 ha (Moore et al. 1999; Waltz et al. 2003). This scale is large enough to address invertebrate (Meyer and Sisk 2001) and vertebrate responses (Germaine and Germaine 2002). Different silvicultural approaches also are being evaluated, including tests of multiple levels of thinning and burn-only treatments (Fulé et al. 2001a, 2001b, 2002b). The degree of thinning was directly related to reduction in potential crown fire behavior, as measured by fire behavior modeling (Scott 1998). Empirical evidence of the effectiveness of thinning in reducing wildfire intensity in Arizona was also reported by Pollet and Omi (2002).

As in other areas of the U.S., the social context of forest restoration has been clouded with disagreement and distrust between some environmental activist groups and governmental agencies such as the U.S. Forest Service. A novel collaboration has attempted to create consensus. The Greater Flagstaff Forests Partnership (2004) was formed after severe wildfires in 1996 by nongovernmental organizations, Northern Arizona University, municipal and county governments, and other stakeholders, to work with the Coconino National Forest to restore forests around the city of Flagstaff, Arizona.

The Partnership and its restoration efforts have not been free of controversy. However, by 2004 the first project was nearly complete, and a series of subsequent projects were also moving forward. The Partnership's work is becoming a useful example of the role of collaboration and its limitations in southwestern forest restoration (Moseley and KenCairn 2001).

31.7.2 Southwestern Colorado — the Ponderosa Pine Partnership

The Ponderosa Pine Partnership in southwestern Colorado began in 1992 with concerns about a dwindling supply of large trees for the local timber industry and overly dense forests of smaller trees vulnerable to destructive wildfires, insect outbreaks, and loss of biodiversity. A public partnership of local national forest administrators, government officials, and representatives of the timber industry sought constructive and sustainable solutions to these problems. A scientific assessment of the ecological history and current state of the landscape (Lynch et al. 2000; Romme et al. 2003) demonstrated natural and human disturbance histories much like those documented in northern Arizona and many other places throughout western North America. Reconstructions of past canopy density in three representative stands ranged from 37 to 59 trees/ha, whereas current densities range from 178 to 338 trees/ha in the same stands (Romme et al. 2003).

The Ponderosa Pine Partnership designed a restoration treatment experiment to test the effectiveness of reaching four goals: reduce canopy density and basal area, leaving clumps of trees interspersed with small openings similar to those of pre-1880 stands; reintroduce low-intensity fire; stimulate productivity and diversity of suppressed herbaceous plants on the forest floor; and provide timber for local sawmills, mostly from small-diameter trees, and to find markets that would allow the companies to harvest small-diameter timber without losing money.

The prescription for achieving these objectives included three stages: retention of the largest and oldest trees and a cluster of trees of all sizes surrounding each large or old tree; fuel reduction with prescribed fire; and ecological and economic monitoring, including a complete accounting of loggers' costs and revenues associated with the project (Romme et al. 2003).

Treatments were conducted from 1995 to 1998, and monitoring was done from 1995 to 2001. The ecological objectives were largely achieved, but monitoring results provided guidelines for improving the results as the project moved forward (Lynch et al. 2000; Romme et al. 2003). Treatments did not significantly reduce the quantity of downed woody fuels on the forest floor, and additional prescribed burning was recommended. The combination of canopy thinning and prescribed burning led to dramatic increases in herbaceous cover and diversity. The economic objective was only partially achieved, the largest obstacle being the lack of suitable markets for small-diameter material (Romme et al. 2003). Numerous creative alternatives for adding value to forest products were explored with only limited success. Overall, the loggers' profit in the entire operation was less than 1% of gross revenues.

In 2001, a scientific review team generally endorsed the approach being taken, and encouraged the partnership to expand restoration treatments from the 20 to 40 ha experimental stands to much larger units. The team recommended creating larger tree clusters and larger open areas than had been accomplished to date, to better emulate the pre-1880 spatial pattern and to provide more appropriate wildlife habitat. The partnership is using these recommendations to develop restoration timber sales for much larger units.

Overall, an ecologically sound restoration prescription has been developed, tested, and refined, and can now be applied more extensively throughout the 75,000-ha ponderosa pine landscape in southwestern Colorado. The local community has supported restoration treatments without legal challenges by environmental or industry groups that often hold up projects of this kind, in large part because of early involvement of researchers and all stakeholder groups in formulating restoration treatments, plus the transparent development and evaluation of the experimental prescription. The largest challenge is financial: the restoration prescription has been only marginally profitable at best. Ultimately, the single greatest economic need for forest restoration projects of this kind is the development of alternative, profitable markets for small-diameter material (Lynch and Mackes 2002). This challenge is being addressed by several institutions, such as the Interior West Center for the Innovative Use of Small Diameter Trees, located at Colorado State University.

31.7.3 East central Colorado — the Upper South Platte Watershed Protection and Restoration Project

The Upper South Platte Project was initiated in the South Platte watershed southwest of Denver, CO, after the 5,000-ha Buffalo Creek fire in 1996. A series of subsequent fires in 2000 and 2002, most spectacularly the 55,000-ha Hayman fire (Graham 2003), contributed to both local and national action on forest restoration. In nearly every case, fire severity was far worse than experienced historically, illustrating a major problem with respect to forest condition in the Colorado Front Range (Kaufmann et al. 2001; Graham 2003). Many structures were burned, and erosion introduced massive amounts of sediment into the metropolitan water supply for Denver, CO. Five firefighters were killed en route to the Hayman fire. Three pilots were killed during aerial suppression activities for a smaller fire near an urban area, and two residents died during postfire flooding.

The continued occurrence of large fires in the Colorado Front Range ponderosa pine zone leads to several conclusions. First, a threshold of forest condition has been reached in which current forests have become particularly vulnerable to large crown fires. Second, there is an urgency to implement restoration and wildland fire hazard mitigation to protect both ecosystems and human values at risk. And third, the scale of treatments must be massive, given the spatial extent of the ponderosa pine zone in the Colorado Front Range and the huge amount of urban encroachment into this zone.

The 1996 Buffalo Creek fire led to the establishment of a partnership to establish goals focused on protecting watersheds and improving the ecological condition of forests. After the 2000 and 2002 fires, a Front Range Fuel Treatment Partnership was formed to address the condition of ponderosa pine forests in the entire Front Range area (several million ha), with an increased focus on the wildland/urban interface.

Research on the historical ecology of Front Range ponderosa pine forests has been a compelling factor guiding restoration efforts. A major portion of this research was conducted on the Cheesman Lake ponderosa pine landscape in the South Platte watershed. While this unlogged 3,000-ha landscape was completely destroyed by the 2002 Hayman fire, it provided exceptional insight into the effects of natural fire behavior and tree recruitment patterns on historical ponderosa pine landscape structure and change over time (Brown et al 1999; Kaufmann et al. 2000b, 2001, 2003; Huckaby et al. 2001, 2003; Fornwalt et al. 2002, 2003). This research demonstrated not only that the ecological condition of current forests was poor, but that characteristics of historical forests could be used to guide treatment selection and implementation.

Initial partners included the U.S. Forest Service (both national forest management and research), Colorado State Forest Service, Denver Water, U.S. Fish and Wildlife Service, U.S. Geological Survey, and U.S. Natural Resource Conservation Service. More recently, environmental organizations have become constructively engaged in moving restoration activities forward. Efforts have been aided by large-scale landscape assessments to identify high-priority watershed and wildland/urban interface areas for initial restoration treatments on both publicly and privately owned land. Public meetings presenting the scientific information available and demonstration sites aided early acceptance of restoration concepts.

From the inception of the restoration project, activities focused on practicing good ecology by altering forest structure toward conditions found before Euro-American settlement. These activities are reducing the probability of crown fires around structures and in watershed areas most critically in need of protection from erosion. Activities also focused on the protection of two endangered species: the Pawnee montane skipper (a butterfly) and the Prebles meadow jumping mouse. Both mechanical thinning and prescribed fire are being used to improve ecological sustainability and mitigate wildfire hazards. Thinning includes both decreasing tree density and canopy cover and creating openings similar to those found historically. This creates fuel breaks that protect property in the wildland/urban interface and limit the size of new crown fires. It also improves butterfly habitat by creating grassy openings. Prescribed fire is focused on reducing fuels where it is safe to do so and burning slash from mechanical treatments; fire will also be used to limit regeneration. Currently, the combined areas of mechanical thinning and prescribed burning treatments are approaching tens of thousands of ha annually. As with other projects, the low economic value of small-diameter trees is a major limitation for large-scale treatment.

31.7.4 Western Montana — the Lick Creek Demonstration Study

In 1982, George Gruell, a pioneer in the use of repeat photography, and his associates published a series of photographs that documented 70 years of management-era change on the structure and composition of ponderosa pine forests (Gruell et al. 1982). They proposed the use of partial cutting and prescribed burning to restore ecological conditions. This led in 1991 to the initiation of the Lick Creek Demonstration Study on the Bitterroot National Forest in west central Montana, documented in a compendium by Smith and Arno (1999). Prior to 1900, light surface fires, believed to have been caused by lightning, burned the area on average every 7 years (Arno 1976; Gruell et al. 1982). The forest consisted primarily of large mature and old-growth pines with grass and forb understory. Based on reconstruction of stand structure, the pines appear to have been predominantly in the 200- to 400-year range (Arno et al. 1995; Menakis 1994).

Management of the Lick Creek site began in 1906. Gifford Pinchot, the first Chief of the U.S. Forest Service (1905 to 1910) and classically trained at the French National Forestry School in Nancy, France, instructed the marking of the trees for cutting (Smith and Arno 1999). The original stand averaged 124 trees per ha. About half the trees were cut favoring larger-diameter trees (trees over 48 cm), and about two thirds of the volume was removed. In the 1950s and 1960s, multiple harvests that favored cutting of the larger pines, combined with fire suppression, led to substantial loss of ecological integrity and a conversion to a stand replacement fire regime (Figure 31.6) (Gruell et al. 1982; Smith and Arno 1999).

In 1991, a series of replicated treatments was located on 215 ha. Three silvicultural systems were tested and combined with varying fuel treatments and controls, and commercial harvesting was conducted in 1992. Treatment effects on trees, understory vegetation, soil nutrients and microbial populations, wildlife forage, and avian use were monitored and evaluated (Smith and Arno 1999). Silvicultural treatments included retention or

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Figure 31.6 Lick Creek restoration site in 1979.

Figure 31.7 Lick Creek restoration site after thinning treatment (U.S. Forest Service file photo).

irregular shelterwood cutting, prescribed for areas that consisted predominantly of second-growth pine 80 to 85 years old, followed by two levels of prescribed burning to treat fuels and rejuvenate understory vegetation; modified individual tree selection cut (Fiedler et al. 1988) with and without prescribed fire on an area of primarily uneven-aged ponderosa pine ranging from seedlings to old-growth trees up to 400 years in age; and commercial thinning in an area where precommercial thinning had taken place in the 1960s, with and without prescribed fire. The cutting and burning prescriptions were designed, in general, to greatly reduce all Douglas fir and to reduce ponderosa pine primarily in the middle- and smaller-size classes. Additional mortality occurred predominantly in the smaller-size classes 2 to 5 years after treatments, due to fire injury and bark beetle attack. The early effects of the treatments on the various response variables are numerous and too complex to describe in this chapter, but are documented by Smith and Arno (1999). However, the treatments were generally successful at achieving the desired stand structure and species composition and at reducing fuels and fire potential in all of the areas (Figure 31.7). Total cover of understory species surpassed pretreatment levels by the second year, particularly in burned treatments. An increase in the presence of invasive weeds and heavy wildlife foraging, particularly in burned areas, compromised some of the restoration goals. Soil studies indicate few significant effects beyond the second year.

The restoration of Lick Creek was not without public controversy. Despite many public hearings and field trips, the treatments were appealed. The long history of documentation of management activities, the availability of repeat photography, and the efforts of a large number of scientists and educators were fundamental to getting this restoration project off the ground. Sound baseline data combined with active monitoring provide a basis for adaptive management as we move forward with restoration projects throughout the region, projects that enjoy increasing public acceptance because of the Lick Creek Demonstration Area.

31.8 Summary

The condition of almost all ponderosa pine forests in the western U.S. is poor, but not irreversibly so, except where recently burned at high severity. While forest densities are high and in many cases there has been an invasion of more fire-sensitive species, there is ample opportunity for ecological restoration and return of ponderosa pine landscapes to an ecologically sustainable condition. The costs of restoration are enormous, however, because fuel treatments to improve ecological conditions and reduce wildfire hazards generally require subsidies in today's economic environment, even when biomass being removed may have some commercial value. Yet, the ecological costs of not treating forests are equally great and often long-lasting because of the nature of current fires. Large crown fires preclude the restoration of sustainable old-growth ponderosa pine forest for centuries, during which many ecological services such as provision for biodiversity are absent or adversely affected.

Evidence is accumulating that restoration activities can return ponderosa pine systems to a condition in which they can once again self-regulate. Restoring the structure of forests and landscapes can set the stage for protection and recovery of sustainable populations of plants and animals. Restoration activities that lead to more open forest structures similar to those found historically result in protection from severe damage during fire because of reduced fire severity, including avoidance of soils destabilized by severe fire. Protection of human lives and property is dramatically easier where crown fires are reduced by an open forest structure. And finally, experience is teaching that humans can adapt more readily than expected to a more open forest landscape structure that had once been considered undesirable, particularly when scientific evidence shows that the open landscape is more natural and ecologically sustainable, and safer for humans.

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Chapter 31/ Restoration of ponderosa pine forests in the interior western U.S. after logging, grazing, and fire suppression

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